J Arid Land (2023) 15(11): 1290–1314
https://doi.org/10.1007/s40333-023-0073-9

Science Press　Springer-Verlag

Evaluation of restoration success in arid rangelands of Iran based on the variation of ecosystem services

Mohsen SHARAFATMANDRAD*, Azam KHOSRAVI MASHIZI

Department of Ecological Engineering, Faculty of Natural Resources, University of Jiroft, Jiroft 7867161167, Iran

Abstract: The plantation of non-native species is one of the most expensive ecological restoration measures in arid and semi-arid areas, while its impacts on local communities are largely ignored. This study assessed the rate of change and the dynamic degree of the economic values of ecosystem services related to local conservation (water yield, stocking rate and aesthetic value) and preserving the future (carbon sequestration, soil protection, soil stability and habitat provision) to determine the restoration success of the plantation of non-native species *Haloxylon ammodendron* (C.A.Mey.) Bunge ex Fenzl (15- and 30-year-old) in parts of arid rangelands of Bardsir region, Kerman Province, Iran. We investigated the impacts of the two plantations on the seven ecosystem services and ecosystem structures (horizontal and vertical structures, vegetation composition and species diversity) based on field sampling and measurements at four sampling sites (i.e., control, degraded, and 15- and 30-year-old plantation sites) in spring and summer of 2022. The restoration success of the plantation of non-native species was then examined by assessing the rate of change and the dynamic degree of the total economic value of all ecosystem services as well as the rate of change and the dynamic degree of the economic values of ecosystem services for the two groups (local conservation and preserving the future). Although the plantation of non-native species *H. ammodendron* enormously improved the vertical and horizontal structures of ecosystems, it failed to increase species diversity and richness fully. Further, despite the plantation of non-native species *H. ammodendron* had significantly increased the economic values of all ecosystem services, it was only quite successful in restoring carbon sequestration. Path analysis showed that plantation age had a significant impact on restoration success directly and indirectly (through changing ecosystem structures and services). The dynamic degree of the economic values of ecosystem services related to local conservation and preserving the future at the 15- and 30-year-old plantation sites indicated that the two plantations successfully restored the ecosystem services related to preserving the future. The presented method can help managers select the best restoration practices and predict their ecological-social success, especially for the plantation of high-risk non-native species in arid and semi-arid areas.

Keywords: *Haloxylon ammodendron*; restoration success; ecosystem services; ecosystem structures; arid ecosystems; path analysis; Iran

Citation: Mohsen SHARAFATMANDRAD, Azam KHOSRAVI MASHIZI. 2023. Evaluation of restoration success in arid rangelands of Iran based on the variation of ecosystem services. Journal of Arid Land, 15(11): 1290–1314. https://doi.org/10.1007/s40333-023-0073-9

1 Introduction

Arid and semi-arid areas cover approximately 42.00% of the Earth's total land area, supporting 38.00% of the world's population, who are often the poor ones (Huang et al., 2017). Land

*Corresponding author: Mohsen SHARAFATMANDRAD (E-mail: mohsen.sharafatmandrad@gmail.com)
Received 2023-04-14; revised 2023-09-16; accepted 2023-09-21
© Xinjiang Institute of Ecology and Geography, Chinese Academy of Sciences, Science Press and Springer-Verlag GmbH Germany, part of Springer Nature 2023

degradation in arid and semi-arid areas is accelerating, and ecosystems are facing serious exploitation problems (Delgado and Marín, 2020). Inappropriate management plans not only cannot lead to improvement, but also may exacerbate land degradation in arid ecosystems (AbdelRahman, 2023). The utter ecological methods should be reconsidered by decision-makers to provide a comprehensive approach to ecosystem management (Ceccon et al., 2020). Sustainable land management is necessary for the health of both ecological and social systems (Löbmann et al., 2022). However, people and the economic benefits they derive from the ecosystems are often ignored in the decision-making process (Chee, 2004). New integrated approaches are needed to bridge the gap between ecological and social assessments of ecological restoration (Löfqvist et al., 2022). Restoration is one of the ultimate goals of ecosystem management, which aims to return degraded ecosystems to the pre-degradation state (Saco et al., 2006). Restoration may change ecosystem structures (Zhou et al., 2020) and functions (Santos et al., 2021). The paucity of information on the impact of these changes is an obstacle to ecosystem restoration (Liniger et al., 2011). Moreover, the revision of approaches in monitoring plant communities' structures and functions is essential for the future decisions of land management (Emamian et al., 2021).

The plantation of plant species adapted to harsh environments is one of the approaches to recovering degraded ecosystems (Vlasenko et al., 2022). Assessment of restoration success is very important for land management due to the high cost of restoration projects (Yıldız et al., 2022). The attributes related to biodiversity (Farahat and Linderholm, 2012), ecosystem structures (Yang et al., 2022) and ecosystem functions (Beldini et al., 2010) are measured to assess the success or failure of plantation projects. New approaches based on ecosystem services are developed for ecological-social assessment of ecosystems in recent years (Peng et al., 2023; Sharafatmandrad and Khosravi Mashizi, 2023). Sustainable restoration projects should provide benefits for both ecological and social systems by improving ecosystem services (Bullock et al., 2011; Keesstra et al., 2018). The economic value of ecosystem services reflects the value of nature (Millennium Ecosystem Assessment, 2005), showing not only the economic importance of ecosystems (Sannigrahi et al., 2020) but also their health status (Yan et al., 2016). Assessing the economic value of ecosystem services under land cover change is an appropriate tool, which can help decision-makers and politicians protect ecosystems and contribute to sustainable development (Liu et al., 2018b). The relationship between land cover change and ecosystem services has been studied widely (Reyers et al., 2009; Ouyang et al., 2016; Gao et al., 2017). Plantation may decrease or increase the supply of ecosystem services by changing vegetation composition (Reyers et al., 2009; Ouyang et al., 2016; Zhang et al., 2016b).

Plant species that conserve ecological integrity and support the well-being of social system should be considered for plantation in arid and semi-arid areas (Reisman-Berman et al., 2019). Both native and non-native species are commonly used for plantation in these areas. However, the plantation of native species does not change ecosystem structures (Wang et al., 2022), while the plantation of non-native species may strongly alter ecosystem structures, functions and processes due to their different growth forms compared to the native ones (Vu Ho et al., 2023). Previous studies have shown both positive and negative impacts of planting non-native species on ecosystems (Bravo et al., 2019; Randriambanona et al., 2019). The plantation of non-native species can alter the biodiversity of ecosystems by changing soil properties, litter decomposition rate, microclimate conditions, etc. (Baker and Murray, 2012). Planting non-native species with thick trunks may reduce the abundance of native species (Harrington and Ewel, 1997). However, non-native species may have positive effects on soil function (Zhu et al., 2020). Randriambanona et al. (2019) showed that afforestation by pine as a non-native species had acted as a catalyst and increased the abundance of native species in the ecosystem. Unfortunately, most studies only focused on tree species and their edaphic impacts (Hoque et al., 2021), timber production and carbon sequestration (Ye Myint et al., 2021), while ignoring other ecosystem services. In addition, monoculture restoration planting is rapidly growing across the world (Liu et al., 2018a).

Reduction in species richness is one of the challenges of monoculture with non-native species (e.g., Rédei et al., 2020; Vu Ho et al., 2023). Planting non-native species requires a comprehensive ecological assessment in order to understand the possible impacts (Reisman-Berman et al., 2019). There is a paucity of information on the impacts of planting non-native species on ecosystem services in arid and semi-arid areas. The plantation of non-native species for ecological restoration needs to proceed with caution, especially in fragile arid ecosystems.

Therefore, a comprehensive approach is needed to evaluate the success of restoration projects in order to assist managers in decision making. In addition to ecological benefits, restoration projects should provide sufficient economic benefits to locals (Zeng et al., 2021). Monitoring the changes of plant communities and the economic value of ecosystem services provides a theoretical reference and a basic scientific method for assessing the success of restoration projects. Current methods of assessing the success of restoration projects often tend to focus on the ecological aspects and neglect the social aspects. In this study, we assessed the variations in seven ecosystem services related to local conservation (water yield, stocking rate and aesthetic value) and preserving the future (carbon sequestration, soil protection, soil stability and habitat provision), as well as ecosystem structures (horizontal and vertical structures, vegetation composition and species diversity), to determine the restoration success of 15- and 30-year-old plantations of non-native species *Haloxylon ammodendron* (C.A.Mey.) Bunge ex Fenzl in parts of arid rangelands of Bardsir region, Kerman Province, Iran. The study pursues the following objectives: (1) the effect of non-native species plantations on ecosystem structures; (2) the effect of non-native species plantations on the economic values of ecosystem services; (3) the relationship between ecosystem structures and ecosystem services; and (4) the assessment of restoration success through assessing the rate of change and the dynamic degree of the economic values of ecosystem services related to local conservation and preserving the future at the 15- and 30-year-old plantation sites. The comprehensive method presented in this study is a step ahead of the previous methods to assess the restoration success of the plantation of non-native species from an ecological-social point of view.

2 Materials and methods

2.1 Study area

This study was carried out in parts of arid rangelands of Bardsir region (29°40′–30°05′N, 56°40′–57°20′E; 1862–2300 m a.s.l.), Kerman Province, southeastern Iran. The landscape is an alluvial plain with deep sandy loam soils that are loose and highly susceptible to wind erosion. The region covers a total area of 9.34×10^4 hm^2, with a mean annual precipitation of 256 mm (which often occurs between October and December). The long-term mean annual temperature is 15.6°C, the mean maximum temperature is 35.0°C (which usually occurs in June and July) and the mean minimum temperature reaches –10.0°C (which usually occurs in January and December).

The dominant plant species of this region is *Artemisia sieberi* Besser., which is severely degraded due to human overexploitation. Non-native species *H. ammodendron* had been planted to restore the ecosystems in the region. *H. ammodendron* is a Mediterranean species that is widespread in Asia and Africa. *Haloxylon* genus is highly adapted to arid and semi-arid areas and can grow well in areas with annual precipitation of less than 100 mm (Wickens et al., 1985). Therefore, this genus is widely used to restore arid and semi-arid areas and stabilize sand dunes due to its high resistance to drought, salinity and heat (Arokh et al., 2021). *H. ammodendron* can obtain water sources differing in space and time by its dimorphic root system (Dai et al., 2022). Branches of *H. ammodendron* have photosynthesis capability, indicating the adaptation of this species to the low humidity and high temperature in drylands (Hu et al., 2021). In Iran, about 2.00×10^6 hm^2 of land area has been afforested by *Haloxylon* sp., especially in central and southeastern regions of the country (Arokh et al., 2021).

2.2 Sampling and measurements of ecosystem structures

In the study area, four sampling sites were selected: control, degraded, and 15- and 30-year-old *H. ammodendron* plantation sites. Sampling sites were similar in terms of topography, soil types and geological formations. At the 15- and 30-year-old *H. ammodendron* plantation sites, *H. ammodendron* plantation projects have been implemented in 2007 and 1992, respectively, to restore vegetation and reduce wind erosion. At the control site, grazing intensity is low (Holechek and Galt, 2000) and the late successional plant species *A. sieberi* is dominant. At the degraded site, the cover of palatable and late successional plant species is reduced due to overgrazing and land use change, and thus annual species *Salsola brachiata* Pall. is dominated.

At each sampling site, the horizontal and vertical structures of ecosystems, vegetation composition and species diversity were measured as ecosystem structures. Landscape Function Analysis (LFA) method was used to evaluate the vertical and horizontal structures of ecosystems. This method was applied in areas with different climate types from dry rangelands in Australia (Tongway and Smith, 1989) to rainforests in Indonesia (Tongway and Hindley, 2003). The accuracy of this method for conditions in Iran has been confirmed by various studies (e.g., Ata Rezaei et al., 2006; Dehghani Bidgoli and Keshavarz, 2018; Sharafatmandrad and Khosravi Mashizi, 2019). For this method, landscapes are divided into patches (perennials and litter) and inter-patches (bare open soil) (Tongway and Hindley, 2004). Three randomly selected 100 m×100 m plots were used to collect data at each sampling site. A 50-m transect was laid out in the direction of the prevailing wind in each plot. The relative area (%) of patches and inter-patches was measured along each transect. Landscape Organization Index (LOI) and canopy volume (m^3/hm^2) of patches were considered as the horizontal and vertical structures of ecosystems, respectively. LOI reflects the proportion of an ecosystem that is able to capture and utilize vital resources (Tongway and Hindley, 2003); it can be calculated by the proportion of the length of patches to the total length of transect (Tongway and Hindley, 2003). Mean canopy area (m^2/hm^2) and height (m) of patches were used to measure the canopy volume of patches in different height classes (Tongway and Hindley, 2003; Saaed et al., 2022). Statistical analyses of data on the length, width and height of patches and the canopy volume in different height classes (Table 1) were performed using LFA method (Tongway and Hindley, 2003). All field sampling and measurements were conducted on May and June of 2022.

Table 1 Height classes of patches (perennials and litter) based on Landscape Function Analysis (LFA) method

Height class	Height (m)	Height class	Height (m)
1	0.00–0.50	4	1.50–2.00
2	0.50–1.00	5	2.00–2.50
3	1.00–1.50	6	2.50–3.00

Simpson's diversity index was used to reflect species diversity as follows:

$$D=\sum_{i=1}^{s}(p_i)^2 , \tag{1}$$

where D is the Simpson's diversity index; s is the plant species; and p_i is the proportion of the number of i^{th} species divided by the total number of all plant species.

2.3 Quantification of the value of ecosystem services

In this study, seven ecosystem services related to local conservation and preserving the future were quantified (Table 2). The Integrated Valuation of Ecosystem Service and Tradeoff (InVEST) model was applied to quantify water yield, carbon sequestration and soil protection. Soil stability, aesthetic value and habitat provision were calculated using field data. Metabolizable energy in forage and metabolic energy required by livestock were applied to estimate the stocking rate at each sampling site (Coupland, 1992; Robles and Passerat, 1995). The detailed description on quantifying the seven ecosystem services is provided in Appendix.

The market price method was used to measure the economic values of carbon sequestration (Ninan and Inoue, 2013), water yield (de Groot et al., 2002) and stocking rate (Baldwin et al., 2022), since there were market prices for these services. The contingent valuation method (CVM) was applied to assess the economic values of non-market services, i.e., soil stability, aesthetic value and habitat provision. CVM is widely used for estimating non-market values (de Groot et al., 2002) and is a stated preference (survey) method that directly asks people about their willingness to pay for a particular ecosystem service (Gürlük, 2006). Previous studies showed that CVM is useful in measuring the economic values of soil stability, habitat provision (de Groot et al., 2002) and aesthetic value (García-Llorente et al., 2012; Rewitzer et al., 2017) of ecosystems. It should be noted that in this study, we converted the economic value of Iranian Rials (IRR) into the international currency (USD) based on the exchange rate in 2022 (1 USD=42,000 Rials).

Table 2 Ecosystem services quantified in this study

Service category	Service type	Reference
Provisioning services	Stocking rate	Havstad et al. (2007)
	Water yield	
Regulating services	Carbon sequestration	Pan et al. (2013)
	Soil protection	
Cultural service	Aesthetic value	van Zanten et al. (2016)
Supporting services	Habitat provision	de Groot et al. (2002); Millennium Ecosystem Assessment (2005)
	Soil stability	

2.4 Determination of restoration success

In this study, we divided the selected seven ecosystem services (water yield, stocking rate, carbon sequestration, soil protection, soil stability, habitat provision and aesthetic value) into two groups based on management goals: local conservation and preserving the future (Himes et al., 2020). Water yield, stocking rate and aesthetic value are services that local people are willing to protect, because these services are directly related to the well-being of locals. Regulating and supporting services that are important for the sustainable production of services in the ecosystem must be preserved as intermediary services for future ecosystem sustainability (Millennium Ecosystem Assessment, 2005). Therefore, the ecosystem services of water yield, stocking rate and aesthetic value were included in local conservation group and the ecosystem services of carbon sequestration, soil protection, soil stability and habitat provision were included in preserving the future group. The restoration success of plantations was then examined by assessing the rate of change and the dynamic degree of the total economic value of all ecosystem services as well as the rate of change and the dynamic degree of the economic values of ecosystem services related to the two groups (local conservation and preserving the future).

The rate of change, which indicates the size and magnitude of the change, was used to measure the changes in the economic values of ecosystem services and ecosystem structures at the degraded, and 15- and 30-year-old *H. ammodendron* plantation sites compared to the control site. It can be calculated as follows (Liu et al., 2018b):

$$R = \frac{\Delta V}{V_r} \times 100\% , \tag{2}$$

$$\Delta V = V_r - V_a , \tag{3}$$

where R (%) is the rate of change; ΔV (USD) is the change in the economic values of ecosystem services (or ecosystem structures); and V_r (USD) and V_a (USD) are the economic values of ecosystem services (or ecosystem structures) at the control site and other sites, respectively.

The dynamic degree of the economic values of ecosystem services is a quantitative value used to compare the economic values of ecosystem services between plantations with different ages and predict their future trends (Liu et al., 2018b). It is calculated as follows:

$$K = \frac{V_d - V_p}{V_d} \times \frac{1}{T} \times 100\%, \tag{4}$$

where K (%) is the dynamic degree of the economic values of ecosystem services; V_p (USD) and V_d (USD) are the economic values of ecosystem services at the plantation and degraded sites, respectively; and T (a) is the plantation age.

Ecosystem structures (horizontal and vertical structures, vegetation composition and species diversity) and services (provisioning services, regulating services, supporting services and cultural service) as well as plantation age were considered as the drivers of restoration success. Path analysis was used to examine the complex relationship between restoration success drivers. It is an advanced statistical method by which, in addition to direct effects, the indirect effects of independent variables on dependent variables can be identified (Lande and Arnold, 1983). Path coefficients can be calculated using the standardized regression coefficients. Regression coefficients are obtained by establishing structural equations, i.e., equations that determine the structure of the assumed relationships in a model (Fan et al., 2016b).

2.5 Statistical analysis

We used one-way analysis of variance (ANOVA) followed by the least significant difference (LSD) to compare the four sampling sites based on data on the economic values of the seven ecosystem services (soil stability, aesthetic value, carbon sequestration, water yield, soil protection, habitat provision and stocking rate) and ecosystem structures (horizontal and vertical structures, vegetation composition and species diversity). Before conducting one-way ANOVA, we checked data for the normality and homogeneity of variance. Pearson's correlation was used to determine the relationship between ecosystem structures and ecosystem services.

3 Results

3.1 Variations in ecosystem structures and ecosystem services

Canopy volume was distributed in three height classes at the control site, with the maximum value of 1200 m^3/hm^2 (Fig. 1). At the degraded site, canopy volume was distributed in three height classes, with the maximum value of 320 m^3/hm^2. At the 15-year-old plantation site, canopy volume was distributed in four height classes, with the maximum value of 932 m^3/hm^2. Further, at the 30-year-old plantation site, canopy volume was distributed in six height classes, with the maximum value of 928 m^3/hm^2.

The relative area of patches with perennials decreased from 41.21% (±1.23%) at the control site to 14.32% (±2.31%) at the degraded site (Table 3). The relative area of patches with perennials at the 15- and 30-year-old plantation sites were 31.20% (±1.32%) and 44.21% (±3.12%), respectively. The LOI values of the control and 15- and 30-year-old plantation sites were 0.86, 0.68 and 0.73, respectively (Table 3). This means that 86.00% of the area at the control site includes patches where resources are conserved, and the other 14.00% of the area at the control site consists of inter-patches where resources may be lost from the ecosystem. For the 15- and 30-year-old plantation sites, 32.00% and 27.00% of the areas consist of inter-patches where resources may be lost from the ecosystem, respectively.

Subshrubs showed the highest percentage of vegetation cover at the control (27.30% (±3.65%)) and degraded (10.25% (±1.32%)) sites (Table 4), while small trees exhibited the highest percentage of vegetation cover at the 15-year-old plantation (25.23% (±5.36%)) and 30-year-old plantation (37.23% (±5.37%)) sites. The highest value of Simpson's diversity index occurred at the control site (0.65 (±0.13)). The Simpson's diversity index values were 0.52 (±0.13) and 0.58 (±0.14) at the 15- and 30-year-old plantation sites, respectively.

Figure 2 shows the changes in ecosystem services (water yield, stocking rate, carbon sequestration, soil protection, habitat provision, soil stability and aesthetic value) at the control, degraded, and 15- and 30-year-old plantation sites. Ecosystem services were improved at the

30-year-old plantation site compared to the 15-year-old plantation site. With the exception of carbon sequestration, all other ecosystem services exhibited the highest and lowest values at the control and degraded sites, respectively. Specifically, carbon sequestration showed the highest value at the 30-year-old plantation site.

The economic value of stocking rate at the control site was 329.20 (±42.54) USD/hm^2, which was higher than the economic values of other ecosystem services (with the exception of carbon sequestration; Table 5). The economic values of carbon sequestration were highest at all sampling sites among all ecosystem services. With the exception of carbon sequestration, the highest economic value of ecosystem services was related to the control site. Moreover, there was no significant difference of economic values between the control and 30-year-old plantation sites in terms of soil stability (P>0.05).

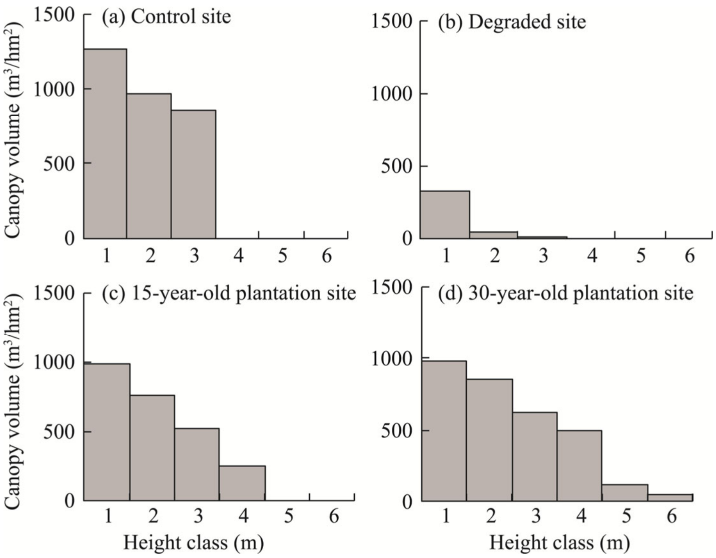

Fig. 1 Distribution of canopy volume in different height classes at the control site (a), degraded site (b), 15-year-old plantation site (c), and 30-year-old plantation site (d)

Table 3 Characteristics of horizontal structure at the control, degraded, and 15- and 30-year-old plantation sites

Horizontal structure		Control site	Degraded site	15-year-old plantation site	30-year-old plantation site
Relative area of patches (%)	Perennials	41.21±1.23^c	14.32±2.31^a	31.20±1.32^b	44.21±3.12^b
	Litter	8.32±3.45^b	1.34±1.65^a	6.23±1.34^b	8.56±2.14^b
Relative area of inter-patches (%)	Bare open soil	50.47±3.12^a	84.34±3.12^c	62.57±3.21^b	47.23±3.12^a
LOI		0.86	0.34	0.68	0.73

Note: Mean±SD. Different lowercase letters within the same row indicate significant differences among the different sampling sites (P<0.05), while the same letter indicates no significant difference.

Table 4 Vegetation cover and species diversity at the control, degraded, and 15- and 30-year-old plantation sites

Index		Control site	Degraded site	15-year-old plantation site	30-year-old plantation site
Vegetation cover (%)	Shrubs	3.56±2.40^a	1.05±0.56^a	1.12±0.56^a	2.14±1.03^a
	Subshrubs	27.30±3.65^b	10.25±1.32^b	1.23±0.35^a	3.13±1.02^a
	Herbs	4.23±0.56^a	3.23±0.35^a	3.21±1.20^a	3.24±1.32^a
	Small trees	0.00±0.00^a	0.00±0.00^a	25.23±5.36^b	37.23±5.37^c
Species diversity	D	0.65±0.13^c	0.21±0.06^a	0.52±0.13^b	0.58±0.14^b
	Richness	26±3^c	8±5^a	12±4^b	13±5^b

Note: D, Simpson's diversity index. Mean±SD. Different lowercase letters within the same row indicate significant differences among the different sampling sites (P<0.05), while the same letter indicates no significant difference.

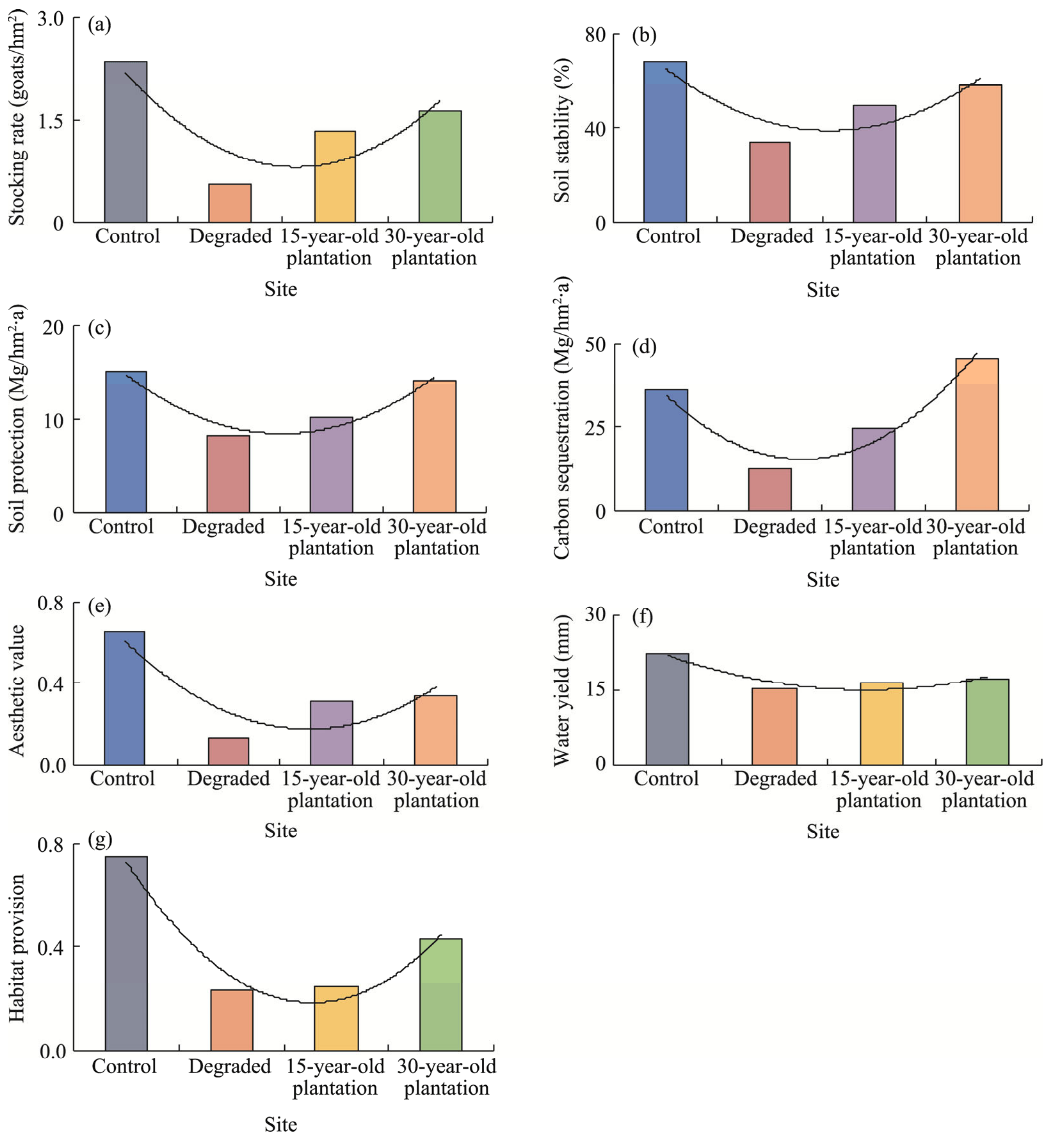

Fig. 2 Variations in ecosystem services of stocking rate (a), soil stability (b), soil protection (c), carbon sequestration (d), aesthetic value (e), water yield (f) and habitat provision (g) at the control, degraded, and 15- and 30-year-old plantation sites

Table 5 Economic values of ecosystem services at the control, degraded, and 15- and 30-year-old plantation sites

Ecosystem service	Economic value (USD/hm^2)			
	Control site	Degraded site	15-year-old plantation site	30-year-old plantation site
Stocking rate	329.20±42.54^d	18.40±2.32^a	89.80±13.54^b	128.40±18.95^c
Water yield	134.03±41.57^c	49.38±8.23^a	98.10±14.23^b	103.38±9.16^b
Carbon sequestration	653.30±85.21^c	228.30±71.70^a	445.10±70.50^b	827.80±98.20^d
Soil protection	12.48±1.41^b	6.74±0.75^a	8.38±1.05^a	11.67±1.06^b
Habitat provision	10.21±1.35^c	3.21±0.23^a	5.35±0.34^b	8.35±0.38^b
Soil stability	14.23±0.35^c	5.68±0.35^a	9.38±1.23^b	13.25±1.54^c
Aesthetic value	18.35±2.68^c	4.21±1.23^a	10.25±3.25^b	12.32±4.39^b

Note: Mean±SD. Different lowercase letters within the same row indicate significant differences among the different sampling sites at $P<0.05$ level, while the same letter indicates no significant difference.

3.2 Determination of the restoration success of 15- and 30-year-old *H. ammodendron* plantations

Figure 3 shows the rates of change in horizontal and vertical structures and species diversity at the degraded, and 15- and 30-year-old plantation sites compared to the control site. The rates of change in horizontal and vertical structures and species diversity were negative at the15-year-old plantation site; while they were positive at the 30-year-old plantation site except for the rate of change in species diversity.

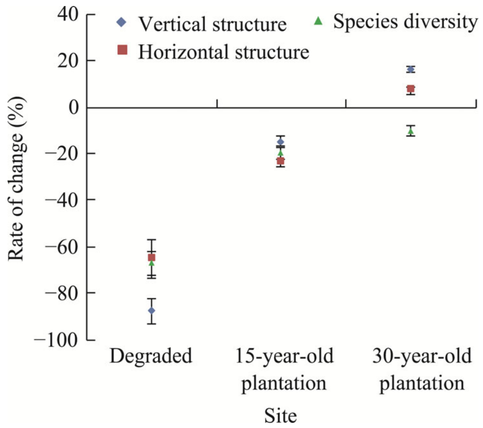

Fig. 3 Rates of change in horizontal and vertical structures and species diversity at the degraded, and 15- and 30-year-old plantation sites compared to the control site. Error bars represent standard deviations.

Comparison of the degraded and plantation sites with the control site showed that the maximum negative rate of change was related to the stocking rate at the degraded site (Table 6). The rates of change in the economic values of the seven ecosystem services were negative but smaller at the two plantation sites than at the degraded site, except for the rate of change of the economic value of carbon sequestration at the 30-year-old plantation site, which was positive and 26.00% higher than that at the control site. In terms of ecosystem services related to the two groups (local conservation and preserving the future) and all ecosystem services, the rates of change were respectively –48.00% (±3.65%), –36.00% (±2.31%), and –42.00% (±3.21%) at the 15-year-old plantation site, and –38.00% (±4.32%), –2.00% (±0.12%) and –20.00% (±5.23%) at the 30-year-old plantation site (Fig. 4).

The dynamic degrees of the economic values of ecosystem services related to the two groups (local conservation and preserving the future) and of the total economic value of all ecosystem services at the 15-year-old plantation site were higher than those at the 30-year-old plantation site (Fig. 5).

Table 6 Rates of change in the economic values of the seven ecosystem services at the degraded, and 15- and 30-year-old plantation sites compared to the control site

Ecosystem service	Rate of change (%)		
	Degraded site	15-year-old plantation site	30-year-old plantation site
Livestock rate	–94.00±2.32	–73.00±12.32	–61.00±8.65
Water yield	–63.17±8.21	–27.00±8.23	–23.00±5.68
Carbon sequestration	–65.78±4.23	–31.00±5.32	26.00±4.35
Soil protection	–46.00±5.65	–32.00±7.65	–06.00±3.20
Habitat provision	–68.00±4.35	–47.00±2.35	–18.00±2.35
Soil stability	–60.00±2.31	–34.00±3.68	–7.00±1.32
Aesthetic value	–77.00±6.23	–44.00±6.98	–32.00±03.68

Note: Mean±SD.

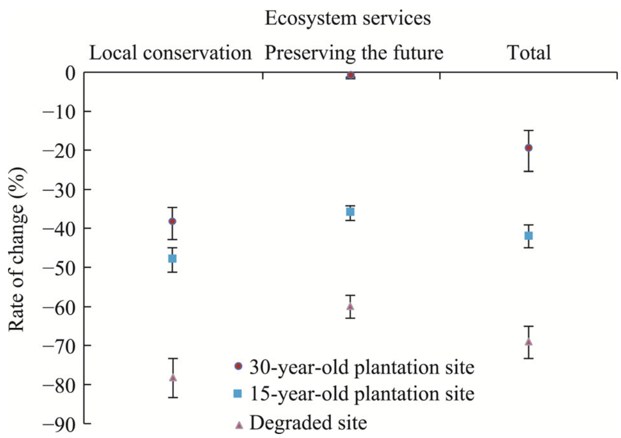

Fig. 4 Rates of change in the economic values of ecosystem services related to the two groups (local conservation and preserving the future) and in the total economic value of all ecosystem services at the degraded, and 15- and 30-year-old plantation sites compared to the control site. Error bars represent standard deviations.

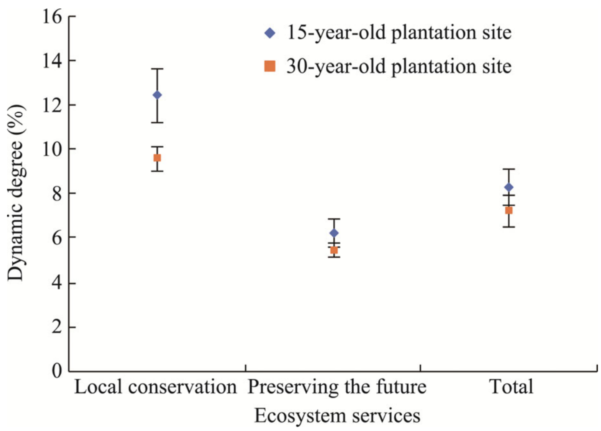

Fig. 5 Dynamic degrees of the economic values of ecosystem services related to the two groups (local conservation and preserving the future) and of the total economic value of all ecosystem services at the 15- and 30-year-old plantation sites compared to the degraded site. Error bars represent standard deviations.

3.3 Impacts of ecosystem services and ecosystem structures as well as plantation age on restoration success

3.3.1 Correlations between ecosystem structures and ecosystem services

Pearson's correlation analysis showed that horizontal structure was significantly positively correlated with water yield, soil protection and habitat provision (P<0.05). Vertical structure was strongly and negatively correlated with stocking rate and positively correlated with carbon sequestration and soil protection (P<0.05). Vegetation composition had a strong negative correlation with carbon sequestration and a strong positive correlation with stocking rate and aesthetic value (P<0.05). Species diversity was positively correlated with stocking rate, water yield, habitat provision and soil stability, and negatively correlated with soil protection (Fig. 6).

3.3.2 Direct and indirect impacts of restoration success drivers

Using path analysis, we investigated the direct and indirect impacts of restoration success drivers (Table 7). The results showed that ecosystem structures, ecosystem services and plantation age all had significant impacts on restoration success. Plantation age had the most direct impact on restoration success (P<0.05). It also indirectly affected restoration success through ecosystem structures and services. The indirect impact of plantation age on restoration success through ecosystem structures was greater than that through ecosystem services (Table 8).

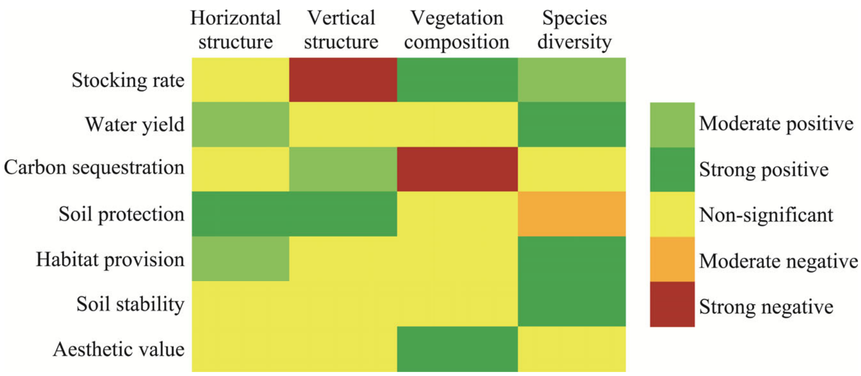

Fig. 6 Correlations between ecosystem structures and ecosystem services

Table 7 Direct and indirect impacts of restoration success drivers based on path analysis

Independent variable	Dependent variable	Standardized beta	t	Adjusted R^2	F
Model 1				0.85	12.35**
Plantation age	Restoration success	0.46	13.24**		
Vertical structure		0.34	8.35**		
Horizontal structure		0.25	3.21*		
Vegetation composition		–0.37	–9.31**		
Species diversity		0.33	7.21**		
Provisioning services		0.23	3.20*		
Regulating services		0.39	10.23**		
Supporting services		0.34	8.35**		
Cultural services		0.27	4.32**		
Model 2				0.83	10.09**
Vertical structure	Plantation age	0.43	9.35**		
Horizontal structure		0.30	5.28**		
Vegetation composition		–0.39	–7.23**		
Species diversity		0.32	6.08**		
Model 3				0.79	7.13**
Provisioning services	Plantation age	0.37	4.85**		
Regulating services		0.47	7.28**		
Supporting services		0.40	5.32**		
Cultural service		0.30	3.68*		

Note: ** indicates significance at the P<0.01 level, and * indicates significance at the P<0.05 level.

Table 8 Standardized impacts of plantation age on restoration success

Restoration success driver	Impact	Standardized beta	t
Plantation age	Direct impact	0.46	13.24**
	Indirect impact through ecosystem structures	0.27	7.67**
	Indirect impact through ecosystem services	0.21	6.34**
	Total impact	0.90	18.34**

Note: ** indicates significance at the P<0.01 level.

4 Discussion

Non-native small tree *H. ammodendron* is very different from native subshrub *A. sieberi*. The plantation of *H. ammodendron* had drastically changed ecosystem structures in arid and semi-arid ecosystems in the region. Changes in ecosystem structures will result in variations of ecosystem services (Assis et al., 2023). Relationships and processes are as important as structural attributes

in ecosystem sustainability (Wu et al., 2022). Thus, the dynamic degree of the economic values of ecosystem services can provide very important information for sustainable restoration projects and ecological development.

4.1 Impact of the plantation of non-native species *H. ammodendron* on ecosystem structures

At the two plantation sites, the plantation of *H. ammodendron* increased the relative area of patches and the height classes of vegetation. There were three more height classes in *H. ammodendron* plantation sites than the control site. Increase of canopy volume in different height classes had a very effective role in promoting the vertical and horizontal structures of ecosystems. *H. ammodendron* plants usually grow in large clumps in arid ecosystems and can enhance the vegetation cover (Ninot et al., 2007). The larger the clumps, the more reduction the environmental disturbances in arid and semi-arid areas (Fryrear, 1995; Marquart et al., 2019). Wind erosion is one of the main disturbances in arid and semi-arid areas (Du et al., 2022). *H. ammodendron* as a natural barrier plays an important role in preventing wind erosion through improving the vertical and horizontal structures of ecosystems. There are complex plant-soil and plant-plant interactions in the ecosystems (Pan et al., 2021). Therefore, the changes in vegetation composition and ecosystem functions resulted from the plantation of non-native species should be considered in ecosystem management.

H. ammodendron plantations had led to the formation of large patches in the ecosystems, thereby affecting the density and diversity of native species in the study area. Only 37.54% of vegetation cover at the degraded site belonged to the dominant subshrubs at the control site. The vegetation cover of subshrubs had decreased to 4.50%–11.46% at the two plantation sites, indicating a strong negative impact of *H. ammodendron* on the dominant native species. Previous studies had also shown that the introduction of non-native species into local ecosystems will affect the composition of native vegetation (e.g., Kotzen, 2003). Some studies stated that monoculture can provide better conditions for native species and lead to higher species diversity (e.g., Cuong et al., 2013). Brown (2003) and El-Wahab et al. (2014) indicated that *Haloxylon* sp. could improve plant diversity of predominantly understory annual species under *H. ammodendron* overstory cover. Elmefregy and El-Sheikh (2020) pointed out that soil properties, hydrological characteristics, dispersal functions and topographic characteristics are the main environmental factors altering the impact of *Haloxylon* sp. on plant diversity. In contrast, some monocultures may prevent the establishment of native seedlings by changing soil properties (Kelty, 2006). *H. ammodendron* affects the composition of native plant communities through changing some physical and chemical properties of soil (Jafari et al., 2004). *Haloxylon* spp. can increase soil salinity (due to salt accumulation in the litter layer) and limit the growth of native plant species (Rathore et al., 2015). On the other hands, *H. ammodendron* has higher competitive ability due to the development of horizontal and vertical roots that usually inhibit the growth of surrounding plants because of allopathic effects (Keneshloo et al., 2018). The higher water absorption capacity of *H. ammodendron* root system is also one of the obstacles to the growth of other plants in arid and semi-arid areas. Zhou et al. (2022) reported the gradual reduction of surface soil moisture over increasing age of *H. ammodendron* plantations in China, and concluded that faster soil drying threatens the growth of smaller native understory plants.

At the 15-year-old plantation site in the study area, richness also decreased by 53.85% compared to the control site. Natural ecosystems have more diverse vegetation composition and age structure than monocultures (Navarro-Cerrillo et al., 2020; Santini et al., 2020). The more plant species there are, the greater the diversity of niches; consequently, the accompanying understory plant species will be more diverse (Larjavaara, 2008). Plantations are more vulnerable to environmental disturbances due to their uniform structures (Camarero et al., 2021).

4.2 Relationship between ecosystem structures and ecosystem services

In the study area, species diversity was the most important attribute of ecosystem structures influencing the supply of some ecosystem services (stocking rate, water yield, habitat provision

and soil stability). Some studies also showed significant positive relationships between species diversity and ecosystem services (Quijas et al., 2010; Dee et al., 2017; Himes et al., 2020). Reale et al. (2022) pointed out that biodiversity loss strongly affects the global economy through changing the supply of ecosystem services. The link between species diversity and ecosystem services showed that biodiversity conservation is a desirable economic solution (Dee et al., 2017).

Provisioning services (water yield and stocking rate) that are directly related to human livelihood are more sensitive to the loss of species diversity (Sharafatmandrad and Khosravi Mashizi, 2021). Díaz et al. (2006) and Schmid et al. (2009) also indicated that plant diversity can increase primary productivity and thus provision of food. Our results showed that supporting services (soil stability and habitat provision) that can provide the basis for supplying other services are strongly related to species diversity. Higher biodiversity is required for the sustainable supply of ecosystem services (Hooper et al., 2005). Increase of species diversity can not only increase the diversity of plant use (Pretzsch and Schütze, 2016), but also contribute to the plant communities that are more resilient against environmental stresses such as drought due to the complementary effects between species with diverse structures (Gazol and Camarero, 2016; Grossiord, 2020).

Diverse ecosystems provide a wider range of services when moderate level of ecosystem functions is desired, but monocultures can maximize the provision of any specific service (van der Plas et al., 2016). In the study area, *H. ammodendron* plantations as monocultures provided a relatively long-time protection against wind erosion through improving the vertical structure of ecosystems, but they failed in increasing species diversity. This may be caused by the failure of *H. ammodendron* plantations in promoting ecosystem multi-functionality. There are both trade-off and synergy relationships between monocultures and the provision of some ecosystem services (Suarez and Gwozdz, 2023). Therefore, restoration programs should be aimed at preserving and enhancing species diversity to sustain the provision of multiple ecosystem services. Some studies showed that plantations with two, three or four plant species are more productive and successful in increasing biodiversity (Liu et al., 2018a; Rivera-Pedroza et al., 2019). Although monoculture is known as the dominant type of plantations in practice (Rivera-Pedroza et al., 2019), it is not different from other diverse ecosystems in providing some ecosystem services. Polley et al. (2020) indicated that the stability of biomass production is similar in monoculture and mixed-species grasslands. Maintaining natural ecosystems alongside monocultures is considered as a strategy to preserve biodiversity in ecosystems where monocultures should be applied (Rivera-Pedroza et al., 2019; Yahya et al., 2022).

4.3 Impact of the plantation of non-native species *H. ammodendron* on ecosystem services

The results of this study showed that 15- and 30-year-old plantations respectively enhanced the total economic value of all ecosystem services 1.23–5.35 and 2.17–7.76 times, compared to the degraded site. Avtar et al. (2014) concluded that plantations can lead to the restoration of ecosystem services such as soil protection, carbon sequestration and water conservation, and may reduce poverty in developing countries through increasing economic benefits. In the study area, the greatest success of plantations was enhanced carbon sequestration (1.8–3.6 times at the plantations sites higher than those at the degraded site). Ahmadi kareh et al. (2013), Loni et al. (2018) and Ma et al. (2021) also indicated that the plantation of *Haloxylon* sp. can enhance the carbon sequestration potential of arid ecosystems. The results of Kumar et al. (2020) also showed that plantations may improve the carbon sequestration of ecosystems in arid and semi-arid areas.

In the study area, ecosystem services were improved at the plantation sites, but this is not the only option for sustainable management. What matters in ecosystem management is how much of the potential of the control site (as an optimal situation in providing ecosystem services) is met by plantations after 15–30 years. Compared to the control site, plantations were only successful in providing carbon sequestration service (Table 6). Previous studies also concluded that *Haloxylon* communities can significantly store more carbon than rangelands by enhancing aboveground and

belowground biomass (Kareh et al., 2013; Ma et al., 2021). Increasing the carbon sequestration potential of ecosystems can modulate the impact of climate change on arid and semi-arid ecosystems (Lal et al., 2011), which is very effective in protecting the future of ecosystems and mitigating the impact of climate change.

At the plantation sites, soil protection service was almost slightly less than that at the control site (Table 6). Previous studies have shown that soil properties may be improved after planting (Singh et al., 2020; Kumi et al., 2021). Soil protection may enhance with increasing silt and clay fractions of the soil in areas planted with *Haloxylon* sp. (Zhang et al., 2016a). Abdi et al. (2019) indicated that *Haloxylon* sp. can increase soil cohesion in arid areas through its root system. Both drought and heavy rainfall may threaten plant species in arid and semi-arid areas (Peñuelas and Filella, 2001), thereby reducing plant growth rate and biomass in ecosystems (Zamin et al., 2013). Soil retention stabilizes plant growth and enhances ecosystem resilience to these threats in arid and semi-arid areas (Wolka, 2014; Zhang et al., 2014). In general, plantations were most successful in providing regulating services and less successful in providing provisioning and cultural services in the study area. Dai et al. (2018) also showed that plantations are more capable of recovering regulating services than other services.

4.4 Impact of plantation age on ecosystem services

The findings of this study showed that the vertical and horizontal structures of ecosystems are improved over increasing plantation age (Fig. 6). Ebrahimi et al. (2017) also indicated that as the age of *H. ammodendron* individuals increases, the cover of plant community is improved, and the density and length of the branches increase, thereby forming larger patches. The number of services provided by ecosystems may also change over increasing plantation age (Zeng et al., 2021).

Regulating, provisioning, supporting and cultural services respectively increased a lot over increasing plantation age. As the age of plantation increases, the amount of carbon stored in ecosystems increases correspondingly (Zhang et al., 2019; Tamang et al., 2021). Older plants can store more carbon due to higher biomass (Fonseca et al., 2012; Wang et al., 2013). In this study, plantation age had a greater impact on provisioning services than supporting services. The dynamic degree of the economic values of ecosystem services did not increase over increasing plantation age. In other words, the dynamic degree of the economic values of ecosystem services at the 15-year-old plantation site was higher than that at the 30-year-old plantation site. The rate of change in ecosystem services was greater in the younger plantations than in the older plantations (Yamaura et al., 2021).

4.5 Restoration success

Restoration planting projects are successful when conservation is done by local people. Public participation in management programs is usually very limited (Shrestha and Shrestha, 2017; Liu et al., 2019). In terms of providing supporting and regulating services that can stabilize ecosystems and are necessary for the future of conservation (Millennium Ecosystem Assessment, 2005), plantations were much more successful in enhancing the potential of ecosystems to provide more regulating services even compared to the control site in the study area. Therefore, plantations were able to enhance the potential of preserving the future of ecosystems and sustain the supply of ecosystem services. Fan et al. (2016a) concluded that *Haloxylon* sp. can restore arid ecosystems by increasing herbaceous plant cover and soil nutrients. Thinning of *H. ammodendron* plantations is proposed to reduce their excessive consumption of soil water (Zhu and Jia, 2011). Restoration of species composition and diversity also plays an important role for ecological success, and our results showed that *H. ammodendron* plantations are not successful in improving species composition and diversity. Hence, the plantation of *H. ammodendron* is relatively success ecologically. Gann et al. (2019) assessed principles for ecological restoration and indicated that ecosystem restoration is a means of improving ecosystems and human well-being in the social system. Improving the provisioning services that are directly related to

human well-being is most important for local people in the social system (Millennium Ecosystem Assessment, 2005). In this study, water yield and stocking rate at the 15- and 30-year-old plantation sites that have direct impacts on social welfare, respectively increased by 2.17 and 5.43 times compared to the degraded site, but they failed to compete over the control site. Therefore, the plantation of *H. ammodendron* cannot be considered successful due to local conservation motivation in the social system. In general, monoculture of *H. ammodendron* will not be successful ecologically and socially. It should be suggested that the combined plantations of *H. ammodendron* with some other plant species that are compatible with *H. ammodendron* can further contribute to the success of restoration in arid ecosystems.

5 Conclusions

The method presented in this study helps managers to evaluate the ecological-social success of restoration in order to select restoration practices, especially plantation with non-native species in arid and semi-arid areas. Land cover changes resulted from plantations first impacted on ecosystem structures. In arid and semi-arid areas that are faced with environmental disturbances such as wind storms, the vertical structure of ecosystems is also an important parameter in addition to the horizontal structure. However, larger species are mostly preferred in plantations in arid and semi-arid areas due to their influences on improving the vertical structure of ecosystems. The results of this study showed that in arid and semi-arid areas, species diversity as an important attribute of ecosystem structures is more effective on improving ecosystem services than vertical structure, which should be considered in management plans in the future. In the process of restoring arid ecosystems, not only the vegetation structure and species interaction ecologically, but also ecosystem services socially should be considered as criteria for species selection. The plantation of non-native species that is structurally different from native species may change all the complex relationships between ecosystem structures and ecosystem services, so this should be considered in sustainable management. Compared to the degraded site, improving the economic values of ecosystem services at the plantation sites does not indicate the success of restoration. The measure of the success should be the degree to which a restoration practice meets the potential of the control site. Sustainable ecosystem management should not only consider ecosystem services related to local conservation to motivate local people to participate in the implementation of restoration projects and protect ecosystems, but also consider ecosystem services related to preserving the future to promote the ecological sustainability of ecosystems. The method presented in this study helps managers to evaluate the ecological-social success of restoration (especially plantations with non-native species in arid and semi-arid areas) to further select restoration practices.

Conflict of interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

This research was supported by the University of Jirof, Iran (4813-00-1). The authors express appreciation for the reviewers and editors who helped improve the quality of this paper.

Author contributions

Conceptualization: Mohsen SHARAFATMANDRAD; Methodology: Mohsen SHARAFATMANDRAD, Azam KHOSRAVI MASHIZI; Formal analysis and investigation: Mohsen SHARAFATMANDRAD, Azam KHOSRAVI MASHIZI; Writing - original draft preparation: Mohsen SHARAFATMANDRAD, Azam KHOSRAVI MASHIZI; Writing - review and editing: Mohsen SHARAFATMANDRAD, Azam KHOSRAVI MASHIZI; Funding acquisition: Mohsen SHARAFATMANDRAD; Resources: Mohsen SHARAFATMANDRAD.

Ethics statement

All experimental protocols were approved by the Review Board of Department of Ecological Engineering, University of Jiroft, Iran. All methods were carried out in accordance with relevant guidelines and regulations. In addition, the participants provided their informed consent to participate in this study.

References

AbdelRahman M A E. 2023. An overview of land degradation, desertification and sustainable land management using GIS and remote sensing applications. Rendiconti Lincei. Scienze Fisiche e Naturali, 34: 767–808.

Abdi E, Saleh H R, Majnonian B, et al. 2019. Soil fixation and erosion control by *Haloxylon persicum* roots in arid lands, Iran. Journal of Arid Land, 11(1): 86–96.

Ahmadi kareh H, Naseri H R, Heshmati G H A. 2013. Comparison of carbon sequestration quantity in *Haloxylon aphyllum* and *Stipagrostis plumosa* in Iran desert area. World Applied Sciences Journal, 21(8): 1190–1193.

Arokh M, Nikbakhat R, Dehghani R, et al. 2021. Investigating the environmental status of *Haloxylon* plantations in Aran va bidgol deserts (Isfahan–Iran). International Archives of Health Sciences, 8(1): 31–36.

Ata Rezaei S, Arzani H, Tongway D. 2006. Assessing rangeland capability in Iran using landscape function indices based on soil surface attributes. Journal of Arid Environments, 65(3): 460–473.

Avtar R, Suzuki R, Sawada H. 2014. Natural forest biomass estimation based on plantation information using PALSAR data. PLoS ONE, 9(1): e86121, doi: 10.1371/journal.pone.0086121.

Baker A C, Murray B R. 2012. Seasonal intrusion of litterfall from non-native pine plantations into surrounding native woodland: Implications for management of an invasive plantation species. Forest Ecology and Management, 277: 25–37.

Baldwin T, Ritten J P, Derner J D, et al. 2022. Stocking rate and marketing dates for yearling steers grazing rangelands: Can producers do things differently to increase economic net benefits? Rangelands, 44(4): 251–257.

Beldini T P, Mcnabb K L, Lockaby B G, et al. 2010. The effect of Amazonian *Eucalyptus* plantations on soil aggregates and organic matter density fractions. Soil Use and Management, 26(1): 53–60.

Bravo S P, Berrondo M O, Cueto V R. 2019. Are small abandoned plantations a threat for protected areas in Andean forests? The potential invasion of non-native cultivated species. Acta Oecologica, 95: 128–134.

Brown G. 2003. Factors maintaining plant diversity in degraded areas of northern Kuwait. Journal of Arid Environments, 54(1): 183–194.

Bullock J M, Aronson J, Newton A C, et al. 2011. Restoration of ecosystem services and biodiversity: conflicts and opportunities. Trends in Ecology & Evolution, 26(10): 541–549.

Camarero J J, Gazol A, Linares J C, et al. 2021. Differences in temperature sensitivity and drought recovery between natural stands and plantations of conifers are species-specific. Science of the Total Environment, 796: 148930, doi: 10.1016/j.scitotenv.2021.148930.

Ceccon E, Méndez-Toribio M, Martínez-Garza C. 2020. Social participation in forest restoration projects: Insights from a national assessment in Mexico. Human Ecology, 48(5): 609–617.

Chee Y E. 2004. An ecological perspective on the valuation of ecosystem services. Biological Conservation, 120(4): 549–565.

Coupland R T. 1992. Ecosystems of the World 8A. Natural Grasslands: Introduction and Western Hemisphere. New York: Elsevier, 1–64.

Cuong C V, Lamb D, Hockings M. 2013. Simple plantations have the potential to enhance biodiversity in degraded areas of Tam Dao National Park, Vietnam. Natural Areas Journal, 33(2): 139–147.

Dai E, Zhu J J, Wang X L, et al. 2018. Multiple ecosystem services of monoculture and mixed plantations: A case study of the Huitong experimental forest of Southern China. Land Use Policy, 79: 717–724.

Dai Y, Wang H W, Shi Q D. 2022. Contrasting plant water-use responses to groundwater depth from seedlings to mature trees in the Gurbantunggut Desert. Journal of Hydrology, 610: 127986, doi: 10.1016/j.jhydrol.2022.127986.

de Groot R S, Wilson M A, Boumans R. 2002. A typology for the classification, description and valuation of ecosystem functions, goods and services. Ecological Economics, 41(3): 393–408.

Dee L E, De Lara M, Costello C, et al. 2017. To what extent can ecosystem services motivate protecting biodiversity? Ecological Letters, 20(8): 935–946.

Dehghani Bidgoli R, Keshavarz A. 2018. Landscape Function Analysis to assess the grazing effect on some soil features in arid ecosystem. Desert Ecosystem Engineering Journal, 7(1): 37–42.

Delgado L E, Marín V H. 2020. Ecosystem services and ecosystem degradation: Environmentalist's expectation? Ecosystem Services, 45: 101177, doi: 10.1016/j.ecoser.2020.101177.

Díaz S, Fargione J, Chapin F S III, et al. 2006. Biodiversity loss threatens human well-being. PLoS Biology, 4(8): e277, doi: 10.1371/journal.pbio.0040277.

Du H Q, Liu X F, Jia X P, et al. 2022. Assessment of the effects of ecological restoration projects on soil wind erosion in northern China in the past two decades. CATENA, 215: 106360, doi: 10.1016/j.catena.2022.106360.

Ebrahimi M, Mohammadi F, Fakhireh A, et al. 2019. Effect of *Haloxylon* spp. with different age classes on vegetation cover and soil properties in an arid desert steppe of Iran. Pedosphere, 29(5): 619–631.

El-Wahab R H A, Al-Rashed A R, Al-Hamad Y. 2014. Conservation condition of *Haloxylon salicornicum* (Moq.) Bunge ex Boiss. in degraded desert habitats of northern Kuwait. International Journal of Current Microbiology and Applied Sciences, 3(10): 310–325.

Elmefregy M, El-Sheikh M A. 2020. Ecological status of sand binder plant white saxaul (*Haloxylon persicum*) at the managed area of Qassim, Saudi Arabia. Applied Ecology and Environmental Research, 18(2): 2781–2794.

Emamian A, Rashki A, Kaskaoutis D G, et al. 2021. Assessing vegetation restoration potential under different land uses and climatic classes in northeast Iran. Ecological Indicators, 122: 107325, doi: 10.1016/j.ecolind.2020.107325.

Fan B L, Zhang A P, Yang Y, et al. 2016a. Long-term effects of xerophytic shrub *Haloxylon ammodendron* plantations on soil properties and vegetation dynamics in northwest China. PLoS ONE, 11(12): e0168000, doi: 10.1371/journal.pone.0168000.

Fan Y, Chen J Q, Shirkey G, et al. 2016b. Applications of structural equation modeling (SEM) in ecological studies: an updated review. Ecological Processes, 5: 19, doi: 10.1186/s13717-016-0063-3.

Farahat E, Linderholm H W. 2012. Ecological impacts of desert plantation forests on biodiversity. African Journal of Ecology, 50(3): 308–318.

Fonseca W, Alice F E, Rey-Benayas J M. 2012. Carbon accumulation in aboveground and belowground biomass and soil of different age native forest plantations in the humid tropical lowlands of Costa Rica. New Forests, 43: 197–211.

Fryrear D W. 1995. Soil losses by wind erosion. Soil Science Society of America Journal, 59(3): 668–672.

Gann G D, McDonald T, Walder B, et al. 2019. International principles and standards for the practice of ecological restoration. Second edition. Restoration Ecology, 27(S1): S1–S46, doi: 10.1111/rec.13035.

Gao J, Li F, Gao H, et al. 2017. The impact of land-use change on water-related ecosystem services: a study of the Guishui River basin, Beijing, China. Journal of Cleaner Production, 163: S148–S155.

García-Llorente M, Martín-López B, Iniesta-Arandia I, et al. 2012. The role of multi-functionality in social preferences toward semi-arid rural landscapes: An ecosystem service approach. Environmental Science & Policy, 19–20: 136–146.

Gazol A, Camarero J J. 2016. Functional diversity enhances silver fir growth resilience to an extreme drought. Journal of Ecology, 104(4): 1063–1075.

Grossiord C. 2020. Having the right neighbors: how tree species diversity modulates drought impacts on forests. New Phytologist, 228(1): 42–49.

Gürlük S. 2006. The estimation of ecosystem services' value in the region of Misi Rural Development Project: Results from a contingent valuation survey. Forest Policy and Economics, 9(3): 209–218.

Harrington R A, Ewel J J. 1997. Invasibility of tree plantations by native and non-indigenous plant species in Hawaii. Forest Ecology and Management, 99(1–2): 153–162.

Havstad K M, Peters D P C, Skaggs R, et al. 2007. Ecological services to and from rangelands of the United States. Ecological Economics, 64(2): 261–268.

Himes A, Puettmann K, Murac B. 2020. Trade-offs between ecosystem services along gradients of tree species diversity and values. Ecosystem Services, 44: 101133, doi: 10.1016/j.ecoser.2020.101133.

Vu Ho K, Kroel-Dulay G, Tolgyesi C, et al. 2023. Non-native tree plantations are weak substitutes for near-natural forests regarding plant diversity and ecological value. Forest Ecology and Management, 531: 120789, doi: 10.1016/j.foreco.2023.120789.

Holechek J I, Galt D. 2000. Grazing intensity guidelines. Rangelands, 22(3): 11–14.

Hooper D U, Chapin F S, Ewel J J, et al. 2005. Effects of biodiversity on ecosystem functioning: a concensus of current knowledge. Ecological Monographs, 75(1): 3–35.

Hoque M Z, Cui S H, Islam I, et al. 2021. Dynamics of plantation forest development and ecosystem carbon storage change in coastal Bangladesh. Ecological Indicators, 130: 107954, doi: 10.1016/j.ecolind.2021.107954.

Hu D, Lv G H, Qie Y D, et al. 2021. Response of morphological characters and photosynthetic characteristics of *Haloxylon ammodendron* to water and salt stress. Sustainability, 13(1): 388, doi: 10.3390/su13010388.

Huang W W, Fonti P, Larsen J B, et al. 2017. Projecting tree-growth responses into future climate: a study case from a Danish-wide common garden. Agricultural and Forest Meteorology, 247: 240–251.

Jafari M, Nik Nahad H, Arfanzadeh R. 2004. Investigating the effects of *Haloxylon aphyllum* on some soil and vegetation

characteristics (Case study: Hossein Abad Region, Qom Province). Desert Journal, 8(1): 152–162. (in Persian)

Kareh H A, Naseri H R, Heshmati G H A. 2013. Comparison of carbon sequestration quantity in *Haloxylon aphyllum* and *Stipagrostis plumose* in Iran Desert Area. World Applied Sciences Journal, 21(8): 1190–1193.

Keesstra S, Nunes J, Novara A, et al. 2018. The superior effect of nature based solutions in land management for enhancing ecosystem services. Science of the Total Environment, 610–611: 997–1009.

Kelty M J. 2006. The role of species mixtures in plantation forestry. Forest Ecology and Management, 233(2–3): 195–204.

Keneshloo M, Nikoo Sh, Kianian M K. 2018. Effects of planting *Haloxylon aphyllum* on carbon sequestration rate and some soil properties in an arid region in Iran. International Journal of Agriculture and Environmental Research, 4(11): 1–7

Kotzen B. 2003. An investigation of shade under six different tree species of the Negev Desert towards their potential use for enhancing microclimatic condition in landscape architectural development. Journal of Arid Environments, 55(2): 231–274.

Kumar R, Bhatnagar P R, Kakade V, et al. 2020. Tree plantation and soil water conservation enhances climate resilience and carbon sequestration of agro ecosystem in semi-arid degraded ravine lands. Agricultural and Forest Meteorology, 282–283: 107857, doi: 10.1016/j.agrformet.2019.107857.

Kumi J A, Kyereh B, Ansong M, et al. 2021. Influence of management practices on stand biomass, carbon stocks and soil nutrient variability of tea plantations in a dry semi-deciduous forest in Ghana. Trees, Forests and People, 3: 100049, doi: 10.1016/j.tfp.2020.100049.

Lal R, Delgado J A, Groffman P M, et al. 2011. Management to mitigate and adapt to climate change. Journal of Soil and Water Conservation, 66(4): 276–285.

Lande R, Arnold S J. 1983. The measurement of selection on correlated characters. Evolution, 37(6): 1210–1226.

Larjavaara M. 2008. A review on benefits and disadvantages of tree diversity. The Open Forest Science Journal, 1: 24–26.

Liniger H, Studer R M, Hauert C, et al. 2011. Sustainable land management in practice: Guidelines and best practices for Sub-Saharan Africa. TerrAfrica: World Overview of Conservation Approaches and Technologies (WOCAT) and Food and Agriculture Organization of the United Nations (FAO).

Liu C L C, Kuchma O, Krutovsky K V. 2018a. Mixed-species versus monocultures in plantation forestry: Development, benefits, ecosystem services and perspectives for the future. Global Ecology and Conservation, 15: e00419, doi: 10.1016/j.gecco.2018.e00419.

Liu X P, Chen X, Hua K P, et al. 2018b. Effects of land use change on ecosystem services in arid area ecological migration. Chinese Geographical Science, 28: 894–906.

Liu Z, Zhang T T, Yu J N, et al. 2019. Determinants of rural households' afforestation program participation: Evidence from China's Ningxia and Sichuan provinces. Global Ecology and Conservation, 17: e00533, doi: 10.1016/j.gecco.2019.e00533.

Löbmann M T, Maring L, Prokop G, et al. 2022. Systems knowledge for sustainable soil and land management. Science of the Total Environment, 822, 153389, doi: 10.1016/j.scitotenv.2022.153389.

Löfqvist S, Kleinschroth F, Bey A, et al. 2022. How social considerations improve the equity and effectiveness of ecosystem restoration. BioScience, 73(2): 134–148.

Loni A, Radnezhad H, Martynova-van Kley A, et al. 2018. The role of *Haloxylon* plantations in improving carbon sequestration potential of sand dunes of Iran. Applied Ecology and Environmental Research, 16(1): 321–333.

Ma Q L, Wang X Y, Chen F, et al. 2021. Carbon sequestration of sand-fixing plantation of *Haloxylon ammodendron* in Shiyang River Basin: Storage, rate and potential. Global ecology and conservation, 28: e01607, doi: 10.1016/j.gecco.2021.e01607.

Marquart A, Eldridge D J, Travers S K, et al. 2019. Large shrubs partly compensate negative effects of grazing on hydrological function in a semi-arid savanna. Basic and Applied Ecology, 38: 58–68.

Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Synthesis. Washington: Island Press, 1–155.

Navarro-Cerrillo R M, Manzanedo R D, Rodriguez-Vallejo C, et al. 2020. Competition modulates the response of growth to climate in pure and mixed *Abies pinsapo* subsp. Maroccana forests in northern Morocco. Forest Ecology and Management, 459: 117847, doi: 10.1016/j.foreco.2019.117847.

Ninan K N, Inoue M. 2013. Valuing forest ecosystem services: Case study of a forest reserve in Japan. Ecosystem Services, 5: 78–87.

Ninot J M, Carrillo E, Font X, et al. 2007. Altitude zonation in the Pyrenees. A geobotanic interpretation. Phytocoenologia, 37(3–4): 371–398.

Ouyang Z Y, Zheng H, Xiao Y, et al. 2016. Improvements in ecosystem services from investments in natural capital. Science, 352(6292): 1455–1459.

Pan S L, Zhang Q M, Meyer-Baese A. 2021. Dynamic analysis of a soil organic matter and plant system with reaction-diffusion. Chaos, Solitons & Fractals, 146: 110883, doi: 10.1016/j.chaos.2021.110883.

Pan Y, Xu Z R, Wu J X. 2013. Spatial differences of the supply of multiple ecosystem services and the environmental and land

use factors affecting them. Ecosystem Services, 5: 4–10.

Peñuelas J, Filella I. 2001. Responses to a warming world. Science, 294(5543): 793–795.

Polley H W, Collins H P, Fay P A. 2020. Biomass production and temporal stability are similar in switchgrass monoculture and diverse grassland. Biomass and Bioenergy, 142: 105758, doi: 10.1016/j.biombioe.2020.105758.

Pretzsch H, Schütze G. 2016. Effect of tree species mixing on the size structure, density, and yield of forest stands. European Journal of Forest Research, 135(1): 1–22.

Quijas S, Schmid B, Balvaner P. 2010. Plant diversity enhances provision of ecosystem services: A new synthesis. Basic and Applied Ecology, 11(7): 582–593

Randriambanona H, Randriamalala J R, Carrière S M. 2019. Native forest regeneration and vegetation dynamics in non-native *Pinus patula* tree plantations in Madagascar. Forest Ecology and Management, 446: 20–28.

Rathore V S, Singh J P, Bhardwaj S, et al. 2015. Potential of native shrubs *Haloxylon salicornicum* and *Calligonum polygonoides* for restoration of degraded lands in arid western Rajasthan, India. Environmental Management, 55(1): 205–216.

Reale R, Ribas L C, Magro Lindenkamp T C. 2022. Ecosystem services as a ballast to guide sustained economic growth by biodiversity conservation actions. Journal of Cleaner Production, 358: 131846, doi: 10.1016/j.jclepro.2022.131846.

Rédei T, Csecserits A, Lhotsky B, et al. 2020. Plantation forests cannot support the richness of forest specialist plants in the forest-steppe zone. Forest Ecology and Management, 461: 117964, doi: 10.1016/j.foreco.2020.117964.

Reisman-Berman O, Keasar T, Tel-Zur N. 2019. Native and non-native species for dryland afforestation: bridging ecosystem integrity and livelihood support. Annals of Forest Science, 76: 114, doi: 10.1007/s13595-019-0903-2.

Rewitzer S, Huber R, Grêt-Regamey A, et al. 2017. Economic valuation of cultural ecosystem service changes to a landscape in the Swiss Alps. Ecosystem Services, 26: 197–208.

Reyers B, O'Farrell P J, Cowling R M, et al. 2009. Ecosystem services, land-cover change, and stakeholders: finding a sustainable foothold for a semiarid biodiversity hotspot. Ecology and Society, 14(1): 38, doi: 10.5751/ES-02867-140138.

Rivera-Pedroza L F, Escobar F, Philpott S M, et al. 2019. The role of natural vegetation strips in sugarcane monocultures: Ant and bird functional diversity responses. Agriculture, Ecosystems & Environment, 284: 106603, doi: 10.1016/j.agee.2019.106603.

Robles A B, Passerat C B. 1995. Native forage shrub species in south-eastern Spain: forage species, forage phytomass, nutritive value and carrying capacity. Journal of Arid Environments, 30(2): 191–196.

Saaed M, Jacobs S, Masubelele M L, et al. 2022. Does the landscape functionality approach provide insight into rangeland conditions in the Tanqua Karoo region, South Africa? PeerJ, 10: e13305, doi: 10.7717/peerj.13305.

Saco P M, Willgoose G R, Hancock G R. 2006. Eco-geomorphology and vegetation patterns in arid and semi-arid regions. Hydrology and Earth System Sciences, 3(4): 2559–2593.

Sannigrahi S, Zhang Q, Joshi P K, et al. 2020. Examining effects of climate change and land use dynamic on biophysical and economic values of ecosystem services of a natural reserve region. Journal of Cleaner Production, 257: 120424, doi: 10.1016/j.jclepro.2020.120424.

Santini F, Shestakova T A, Dashevskaya S, et al. 2020. Dendroecologicaland genetic insights for future management of an old-planted forest of the endangered Mediterraneanfir fir *Abies pinsapo*. Dendrochronologia, 63: 125754, doi: 10.1016/j.dendro.2020.125754.

Santos F M, Terra G, Piotto D, et al. 2021. Recovering ecosystem functions through the management of regenerating community in agroforestry and plantations with *Khaya* spp. in the Atlantic Forest, Brazil. Forest Ecology and Management, 482: 118854, doi: 10.1016/j.foreco.2020.118854.

Schmid B, Pfisterer A B, Balvanera P. 2009. Effects of biodiversity on ecosystem, community, and population variables reported 1974–2004. Ecology, 90(3): 853, doi: 10.1890/08-1465.1.

Sharafatmandrad M, Khosravi Mashizi A. 2019. Efficacy of landscape function analysis to assess differences between grazed and ungrazed rangelands in an arid landscape. Range Management and Agroforestry, 40(2): 196–201.

Shrestha S, Shrestha U B. 2017. Beyond money: Does REDD+payment enhance household's participation in forest governance and management in Nepal's community forests? Forest Policy and Economics, 80: 63–70.

Singh A K, Sahu C, Sahu S K. 2020. Carbon sequestration potential of a teak plantation forest in the Eastern Ghats of India. Journal of Environmental Biology, 41(4): 770–775.

Suarez A, Gwozdz W. 2023. On the relation between monocultures and ecosystem services in the Global South: A review. Biological Conservation, 278: 109870, doi: 10.1016/j.biocon.2022.109870.

Tamang M, Chettri R, Vineeta, et al. 2021. Stand structure, biomass and carbon storage in Gmelina arborea plantation at agricultural landscape in foothills of Eastern Himalayas. Land, 10(4), 387, doi: 10.3390/land10040387.

Tongway D J, Smith E L. 1989. Soil surface features as indicators of rangeland site productivity. Australian Rangeland Journal. 11(1): 15–20.

Tongway D J, Hindley N. 2003. Indicators of ecosystem rehabilitation success. Stage Two. Verification of EFA Indicators. Final Report for the Australian Center for Mining Environmental Research. Canberra: CSIRO Sustainable Ecosystems, 1–66.

van der Plas F, Manning P, Allan E, et al. 2016. Jack-of-all-trades effects drive biodiversity-ecosystem multifunctionality relationships in European forests. Nature Communications, 7(1): 11109, doi: 10.1038/ncomms11109.

van Zanten B T, Zasada I, Koetse M J, et al. 2016. A comparative approach to assess the contribution of landscape features to aesthetic and recreational values in agricultural landscapes. Ecosystem Services, 17: 87–98.

Vlasenko M V, Rybashlykova L P, Turko S Y. 2022. Restoration of degraded lands in the arid zone of the European part of Russia by the method of phytomelioration. Agriculture, 12(3): 437, doi: 10.3390/agriculture12030437.

Wang F M, Xu X, Zou B, et al. 2013. Biomass accumulation and carbon sequestration in four different aged *Casuarina equisetifolia* coastal shelterbelt plantations in South China. PLoS ONE, 8: e77449, doi: 10.1371/journal.pone.0077449.

Wang T, Dong L B, Liu Z G. 2022. Factors driving native tree species restoration in plantations and tree structure conversion in Chinese temperate forests. Forest Ecology and Management, 507: 119989, doi: 10.1016/j.foreco.2021.119989.

Wickens D E, Goodin J R, Field D V. 1985. Plants for Arid Lands. Cambridge: Cambridge University Press, 1–452.

Wolka K. 2014. Effect of soil and water conservation measures and challenges for its adoption: Ethiopia in focus. Journal of Environmental Science and Technology, 7(4): 185–199.

Wu H, Hu B, Han H, et al. 2022. Network analysis reveals the regulatory effect of mixed stands on ecosystem structure and functions in the Loess Plateau, China. Science of the Total Environment, 824: 153588, doi: 10.1016/j.scitotenv.2022.153588.

Yahya M S, Atikah S N, Mukri I, et al. 2022. Agroforestry orchards support greater avian biodiversity than monoculture oil palm and rubber tree plantations. Forest Ecology and Management, 513: 120177, doi: 10.1016/j.foreco.2022.120177.

Yamaura Y, Yamada Y, Matsuura T, et al. 2021. Modeling impacts of broad-scale plantation forestry on ecosystem services in the past 60 years and for the future. Ecosystem Services, 49: 101271, doi: 10.1016/j.ecoser.2021.101271.

Yan Y, Zhao C L, Wang C X, et al. 2016. Ecosystem health assessment of the Liao River Basin upstream region based on ecosystem services. Acta Ecologica Sinica, 36(4): 294–300.

Yang B Y, Ali A, Xu M S, et al. 2022. Large plants enhance aboveground biomass in arid natural forest and plantation along differential abiotic and biotic conditions. Frontiers in Plant Science, 13: 999793, doi: 10.3389/fpls.2022.999793.

Ye Myint Y, Sasaki N, Datta A, et al. 2021. Management of plantation forests for bioenergy generation, timber production, carbon emission reductions, and removals. Cleaner Environmental Systems, 2: 100029, doi: 10.1016/j.cesys.2021.100029.

Yıldız O, Eşen D, Sargıncı M, et al. 2022. Restoration success in afforestation sites established at different times in arid lands of Central Anatolia. Forest Ecology and Management, 503: 119808, doi: 10.1016/j.foreco.2021.119808.

Zamin N T, Machado S A, Filho A F, et al. 2013. Effect of climate variables on monthly growth in modeling biological yield of *Araucaria angustifolia* and *Pinus taeda* in the juvenile phase. International Journal of Environmental Research, 2013: 646759, doi: 10.1155/2013/ 646759.

Zeng Y L, Wu H L, Ouyang S A, et al. 2021. Ecosystem service multifunctionality of Chinese fir plantations differing in stand age and implications for sustainable management. Science of the Total Environment, 788: 147791, doi: 10.1016/j.scitotenv.2021.147791.

Zhang F, Xing Z S, Rees H W, et al. 2014. Assessment of effects of two runoff control engineering practices on soil water and plant growth for afforestation in a semi-arid area after 10 years. Ecological Engineering, 64: 430–442.

Zhang K, Su Y Z, Wang T, et al. 2016a. Soil properties and herbaceous characteristics in an age sequence of *Haloxylon ammodendron* plantations in an oasis-desert ecotone of northwestern China. Journal of Arid Land, 8(6): 960–972.

Zhang X, Zhang X L, Han H, et al. 2019. Biomass accumulation and carbon sequestration in an age-sequence of Mongolian pine plantations in Horqin Sandy Land, China. Forests, 10(2): 197, doi: 10.3390/f10020197.

Zhang Y, Zhang C B, Wang Z Q, et al. 2016b. Vegetation dynamics and its driving forces from climate change and human activities in the Three-River Source Region, China from 1982 to 2012. Science of the Total Environment, 563–564: 210–220.

Zhou D M, Si J H, He X H. et al. 2022. The process of soil desiccation under *Haloxylon ammodendron* plantations: A case study of the Alxa Legue Desert, China. Plants, 11(3): 235, doi: 10.3390/plants11030235.

Zhou J J, Zhao Y R, Huang P, et al. 2020. Impacts of ecological restoration projects on the ecosystem carbon storage of inland river basin in arid area, China. Ecological Indicators, 118: 106803, doi: 10.1016/j.ecolind.2020.106803.

Zhu Y, Wang Y F, Chen L D. 2020. Effects of non-native tree plantations on soil microarthropods and their feeding activity on the Chinese Loess Plateau. Forest Ecology and Management, 477: 118501, doi: 10.1016/j.foreco.2020.118501.

Zhu Y J, Jia Z Q. 2011. Soil water utilization characteristics of *Haloxylon ammodendron* plantation with different age during summer. Acta Ecologica Sinica, 31(6): 341–346.

Appendix

The detailed information on the quantification of the seven ecosystem services is as follows.

1 Soil stability

To measure soil stability, we derived soil stability index (%) using the Landscape Function Analysis (LFA) methodology. For each patch identified along the transect, seven soil surface indicators were assessed: soil cover, litter cover, cryptogam cover, crust broken-ness, erosion type and severity, deposited materials and surface resistance to disturb and slake (Tongway and Hindley, 2003). The contingent valuation method (CVM) was used to estimate the economic value of soil stability.

Supporting services indirectly benefit people by supporting other services (Millennium Ecosystem Assessment, 2005). Therefore, the indirect benefits of soil stability (for services such as water and livestock production) were explained by the respondents connected with nature and living around the sampling sites. The field investigation was conducted on May and June of 2022, and respondents (180 individuals) were asked "Do you agree with the benefits demonstrated and how much are you willing to pay for enhancing these benefits (supplying of soil stability)?".

2 Aesthetic value

Two 100-m transects were laid out at each sampling site. Thirty 2 m×2 m quadrats were randomly established along each transect. In each quadrat, canopy cover (%), litter cover (%), and stone and gravel percentage (%) were estimated. The number of plant individuals was also recorded. Previous researches showed that people's perception of aesthetic value can be enhanced by the abundance of flowering plants, as well as their colorfulness and blooming seasons (Graves et al., 2017a, b; Uchida et al., 2020). Using the following equation, the aesthetic value for each quadrat was estimated (Schirpke et al., 2017):

$$\mathrm{AV} = \mathrm{r}D+rFLP+rFCA+rFPCC\ , \tag{1}$$

where AV is the aesthetic value; rD is the relative Simpson's diversity index; $rFLP$ is the relative flowering period; $rFCA$ is the relative amount of flower colors; and $rFPCC$ is the relative canopy cover of flowering plants. The parameter of rD was estimated by dividing the Simpson's diversity index of each quadrat by the maximum Simpson's diversity index of all quadrats; $rFLP$ was obtained by dividing the total flowering time of the species in each quadrat by the total flowering time of all existing species in all quadrats; $rFCA$ was estimated by dividing the maximum number of flower colors observed in each quadrat by the number of flower colors in all quadrats; and $rFPCC$ was calculated by dividing the canopy cover of flowering plants in each quadrat by the maximum canopy cover of flowering plants in all quadrats (Schirpke et al., 2017).

We determined the flowering period of plants according to literature (Karimi et al., 2008; Toopchi-Khosroshahi and Lotfalizadeh, 2011; Ariapour et al., 2015). We also checked the flower colors observed in the quadrats based on botanical literature (Rechinger, 1997). CVM was used to estimate the economic value of aesthetic value. A sample size of 180 individuals was obtained using Cochran method (Cochran, 1997). Respondents (180 individuals) were randomly selected from visitors and people who are connected with nature and lived around the sampling sites. A questionnaire method was applied to assess people's willingness to pay for aesthetic value. Respondents were asked "How much are you willing to pay for a day or more on this site if you enjoy these views" based on the method of García-Llorente et al. (2012).

3 Carbon sequestration

The Integrated Valuation of Ecosystem Service and Tradeoff (InVEST) model was used to

measure carbon sequestration. In this model, aboveground biomass carbon (Mg/hm^2), underground biomass carbon (Mg/hm^2), litter carbon (Mg/hm^2) and soil carbon (Mg/hm^2) can be estimated. Thirty 2 m×2 m quadrats were randomly established at the control and degraded sites. For the two plantation sites, thirty 10 m×10 m quadrats were randomly established. In each quadrat, the canopy cover (%), height (m) and number of individuals of plant species were determined. Double sampling method was used to estimate the aboveground biomass (Mg/hm^2) of the species (Reid et al., 1990). Hence, the aboveground biomass was visually estimated in each quadrat. However, a quarter of the quadrats were considered as calibrated quadrats, in which the aboveground biomass was also measured by direct harvesting (clipping and weighing) after estimation. The clipped samples were transferred to the laboratory to determine the moisture content and carbon content. The root/stem ratio (the ratio of the underground biomass to the aboveground biomass) was applied to estimate the underground biomass of the species. For this purpose, five average-size individuals were selected for each plant species and their root samples were taken by digging a soil pit to the rooting depth, and then the samples were weighted. A total of 100 g of root samples were taken to determine the moisture content and carbon content. Ten soil samples were randomly taken from 0–30 cm soil depths at each sampling site. The organic matter of all samples (i.e., aboveground and belowground biomass, litter and soil organic matter) was determined by wet and dry combustion. Finally, 11.00 USD was considered as carbon price per ton to estimate the economic value of carbon sequestration (Dang et al., 2022).

4 Water yield

Water yield was quantified using the water yield model in the InVEST model. The water yield model has been applied in different regions of Europe (Bangash et al., 2013), America (Hamel et al., 2015), Africa (Belete et al., 2020) and Iran (Daneshi et al., 2021; Sharafatmandrad and Khosravi Mashizi, 2021; Balist et al., 2022; Darvishi and Yousefi, 2022) to quantify water yield.

In this model, water yield was estimated by subtracting evapotranspiration from precipitation in the region (Tallis et al., 2011). Required data in this model are as follows: precipitation (mm), reference evapotranspiration (mm), plant water availability (cm^3/cm^3), soil depth (m), crop coefficient and maximum rooting depth (m). The Penman-Monteith equation was used to estimate reference evapotranspiration based on daily mean, maximum and minimum temperatures obtained from local weather stations. Plant water availability was determined using soil texture data (De Ridder and van Keulen, 1995). Soil depth was estimated using the methodology of Tsai et al. (2001), Tesfa et al. (2009) and Liu et al. (2013) based on topography characteristics determined by digital elevation model (DEM) and soil cover. Crop coefficient was estimated according to the method of Allen-Wardell et al. (1998) and leaf area index was determined using the gravimetric method (Jonckheere et al., 2004). The maximum rooting depth was calculated according to the studies of Canadell et al. (1996), Schenk and Jackson (2002) and Tallis et al. (2011). Water price per cubic meters was considered to be 0.60 USD (Mousavi et al., 2021).

5 Soil protection

The InVEST model was used to estimate soil protection. This model can be applied to reveal the impacts of land cover change on soil protection (Hamel et al., 2015), which has been tested in many countries and regions of the world, e.g., Ethiopia (Aneseyee et al., 2019), China (Zhou et al., 2019), Tana watershed, Kenya (Ayana et al., 2017), USA (Hamel et al., 2015) and Iran (Khosravi Mashizi et al., 2019). In this model, soil loss can be estimated using the universal soil loss equation (USLE) (Renard et al., 1997). Soil protection (S; Mg/(hm^2•a)) was estimated by soil loss from non-vegetated areas ($Sl_{\max}$; Mg/(hm^2•a)) minus the actual soil loss (SL_j; Mg/(hm^2•a)) as follows:

$$S = Sl_{\max} - SL_j , \quad (2)$$

$$Sl_{\max} = R \times K \times L \times S \times P , \quad (3)$$

$$SL_j = R \times K \times L \times S \times C \times P, \tag{4}$$

where R (MJ·mm/(hm^2·h·a)) is the total rainfall erosivity in one year; K (t·hm^2/(MJ·mm)) is the soil erodiblity factor; L and S are the slope length and steepness, respectively; P is the erosion control practice factor; and C is the vegetation management factor. R was estimated using the method of Wischmeier and Smith (1978). K was calculated using the method of Romkens et al. (1986) based on soil texture. Soil samples were randomly taken from 0–25 cm soil depths at each sampling site to determine soil texture. The hydrometer method was used to determine the particle size distribution (sand, silt and clay contents) of each soil sample (Soil Survey Staff, 1994). L and S were estimated using the DEM data based on the method of McCool et al. (1987). P was estimated using the Wener method (Lufafa et al., 2003), and C was determined based on the study of Wischmeier and Smith (1978). Soil price per ton was considered to be 0.82 USD (Rastgar et al., 2016).

6 Habitat provision

To estimate the diversity of subsoil macrofauna, we established ten 25 cm×25 cm×25 cm plots under the dominant shrubs at each sampling site, and soil samples were taken (Palacios-Vargas et al., 2007; Sandhu et al., 2010; Ghaley et al., 2014). Macrofauna of each soil sample were separated by hand and killed by immersion in ethyl alcohol (70%). Macrofauna were identified at the family level. The diversity of understory plant species was also measured in 30 quadrats (50 cm×50 cm) established under the dominant shrubs at each sampling site (Bueno et al., 2013). Simpson's diversity index was used to calculate the diversity of understory plant species and macrofauna. The average value of Simpson's diversity index for the diversity of understory plant species and macrofauna was considered as the potential of habitat provision. CVM was used to estimate the economic value of habitat provision. Habitat provision as a supporting service indirectly provides benefits for people. Like soil stability service, the benefits of habitat provision were explained and respondents (180 individuals) were asked "Do you agree with the benefits demonstrated and how much are you willing to pay for enhancing the benefits (supplying of habitat provision)?".

7 Stocking rate

Forage supply (stocking rate) is the main provisioning service of rangelands (Oñatibia et al., 2015). It is one of the main services of rangeland ecosystems, and is drastically depends on management plans (Maes et al., 2012). Metabolizable energy in forage and metabolic energy required by livestock were used to estimate the stocking rate at each sampling site (Coupland, 1992; Robles and Passera, 1995). Metabolizable energy in forage was determined using forage quantity and quality. Thirty 2 m×2 m quadrats were randomly placed at the control and degraded sites. For each of the two plantation sites, thirty 10 m×10 m quadrats were randomly established. The amount of forage produced in each quadrat was measured by double sampling method. Metabolizable energy of forage was also estimated by determining nitrogen content (%) and acid detergent fiber (ADF; %) of forage in each quadrat (Minson, 1987). Nitrogen content of samples was determined by Kjeldahl method (Sáez-Plaza et al., 2013), and ADF was measured using the method of Van Soest et al. (1991). Daily required metabolic energy by livestock energy requirement was also estimated using the Ministry of Agriculture, Fisheries & Food (MAFF) method (MAFF, 1975). Goat (average weight of 30–35 kg) is the dominant livestock in the study area. The economic value of stocking rate for each sampling site was estimated using the market price of meat.

References

Allen-Wardell G, Bernhardt P, Bitner R, et al. 1998. The potential consequences of pollinator declines on the conservation of biodiversity and stability of food crop yields. Conservation Biology, 12(1): 8–17.

Aneseyee A B, Elias E, Soromessa T, et al. 2019. Land use/land cover change effect on soil erosion and sediment delivery in the

Winike watershed, Omo Gibe Basin, Ethiopia. Science of the Total Environment, 728: 138776, doi: 10.1016/j.scitotenv.2020.138776

Ariapour A, Mehrabi H R, Kheradmand G. 2015. Evaluating range plant species suitability for apiculture (Case study: rangeland Sarab Sefid, Boroujerd, Lorestan). Journal of Rangeland, 9: 142–158.

Ayana E K, Fisher J R B, Hamel P, et al. 2017. Identification of ditches and furrows using remote sensing: application to sediment modelling in the Tana watershed, Kenya. International Journal of Remote Sensing, 38(16): 4611–4630.

Balist J, Malekmohammadi B, Jafari H R, et al. 2022. Modeling the supply, demand, and stress of water resources using ecosystem services concept in Sirvan River Basin (Kurdistan–Iran). Water Supply, 22(3): 2816–2831.

Bangash R F, Passuello A, Sanchez-Canales M, et al. 2013. Ecosystem services in Mediterranean river basin: climate change impact on water provisioning and erosion control. Science of the Total Environment, 458–460: 246–255.

Belete M, Deng J, Abubakar G A, et al. 2020. Partitioning the impacts of land use/land cover change and climate variability on water supply over the source region of the Blue Nile Basin. Land Degradation & Development, 31(15): 2168–2184.

Bueno C G, Azorín J, Gómez-García D, et al. 2013. Occurrence and intensity of wild boar disturbances, effects on the physical and chemical soil properties of alpine grasslands. Plant and Soil, 373: 243–256.

Canadell J, Jackson R B, Ehleringer J B, et al. 1996. Maximum rooting depth of vegetation types at the global scale. Oecologia, 108(4): 583–595.

Cochran W G. 1977. Sampling Techniques (3rd ed.). New York: John Wiley & Sons, 1–428.

Coupland R T. 1992. Ecosystems of the World 8A. Natural Grasslands: Introduction and Western Hemisphere. New York: Elsevier, 1–64.

Daneshi A, Brouwer R, Najafinejad A, et al. 2021. Modelling the impacts of climate and land use change on water security in a semi-arid forested watershed using InVEST. Journal of Hydrology, 593: 125621, doi: 10.1016/j.jhydrol.2020.125621.

Dang K B, Phan T T H, Nguyen T T, et al. 2022. Economic valuation of wetland ecosystem services in northeastern part of Vietnam. Knowledge & Management of Aquatic Ecosystems, 423: 12, doi: 10.1051/kmae/2022010.

Darvishi A, Yousefi M. 2022. Using water yield ecosystem services to assess water scarcity in a metropolitan arid environment in Qazvin region (Iran). [2023-03-20]. https://raco.cat/index.php/PapersIERMB/index.

De Ridder N, van Keulen H. 1995. Estimating biomass through transfer functions based on simulation model results: a case study for Sahel. Agricultural Water Management, 28(1): 57–71.

García-Llorente M, Martín-López B, Iniesta-Arandia I, et al. 2012. The role of multi-functionality in social preferences toward semi-arid rural landscapes: An ecosystem service approach. Environmental Science & Policy, 19–20: 136–146.

Ghaley B B, Vesterdal L, Porter J R. 2014. Quantification and valuation of ecosystem services in diverse production systems for informed decision-making. Environmental Science & Policy, 39: 139–149.

Graves R A, Pearson S M, Turner M G. 2017a. Landscape dynamics of floral resources affect the supply of a biodiversity-dependent cultural ecosystem service. Landscape Ecology, 32: 415–428.

Graves R A, Pearson S M, Turner M G. 2017b. Species richness alone does not predict cultural ecosystem service value. Proceeding of the National Academy of Science USA, 114(14): 3774–3779.

Hamel P, Chaplin-Kramer R, Sim S, et al. 2015. A new approach to modeling the sediment retention service (InVEST 3.0): Case study of the Cape Fear catchment, North Carolina, USA. Science of the Total Environment, 524–525: 166–177.

Jonckheere I, Fleck S, Nackaerts K, et al. 2004. Review of methods for in situ leaf area index determination: Part I. Theories, sensors and hemispherical photography. Agricultural and Forest Meteorology, 121(1–2): 19–35.

Karimi A H, Nazarian H, Jafari E. 2008. Identification of Fars honey bee plant resources from three families in Fars Province (Asteraceae, Papilionaceae and Lamiaceae). Pajouhesh and Sazandegi, 2(75): 101–111. (in Persian)

Khosravi Mashizi A, Heshmati G A, Salman Mahini A R, et al. 2019. Exploring management objectives and ecosystem service trade-offs in a semi-arid rangeland basin in southeast Iran. Ecological Indicators, 98: 794–803.

Liu J T, Chen X, Lin H, et al. 2013. A simple geomorphic-based analytical model for predicting the spatial distribution of soil thickness in headwater hillslopes and catchments. Water Resources Research, 49(11): 7733–7746.

Lufafa A, Tenywa M M, Isabirye M, et al. 2003. Prediction of soil erosion in a Lake Victoria basin catchment using a GIS-based Universal Soil Loss model. Agricultural Systems, 76(3): 883–894.

Maes J, Paracchini M L, Zulian G, et al. 2012. Synergies and trade-offs between ecosystem service supply, biodiversity, and habitat conservation status in Europe. Biological Conservation, 155: 1–12.

McCool D K, Brown L C, Foster G R, et al. 1987. Revised slope steepness factor for the universal soil loss equation. Transactions of the ASABE, 30(5): 1387–1396.

Millennium Ecosystem Assessment. 2005. Ecosystems and Human Well-being: Synthesis. Washington: Island Press, 1–155.

Ministry of Agriculture, Fisheries & Food (MAFF). 1975. Energy allowances and feeding systems for ruminants. In: Technical Bulletin, Ministry of Agriculture, Fisheries and Food. London: HM Stationery Office, 1–79.

Minson D J. 1987. Estimation of the nutritive value of forage. In: Wheeler J L, Pearson C J, Roberts G E. Temperate Pastures:

Their Production, Use and Management. An Australian Wool Corporation Technical Publication. Melbourne: Australian Wool Corporation and CSIRO, 415–422.

Mousavi S M, Sarai Tabrizi M, Talachi Langeroudi H. 2021. Investigating the economic value of water in environmental, agricultural and industrial uses (Case Study: Urmia Lake Watershed). Human and Environment, 19(3): 79–95.

Oñatibia G R, Aguiar M R, Semmartin M. 2015. Are there any trade-offs between forage provision and the ecosystem service of C and N storage in arid rangelands? Ecological Engineering, 77: 26–32.

Palacios-Vargas J G, Castaño-Meneses G, Gómez-Anaya J A, et al. 2007. Litter and soil arthropods diversity and density in a tropical dry forest ecosystem in Western Mexico. Biodiversity and Conservation, 16: 3703–3717.

Rastgar S, Barani H, Darijani A, et al. 2016. Estimating direct economic value of soil conservation function of rangelands vegetation (Case study: summer rangelands of Nour-Rud Watershed Basin). Journal of Watershed Management Research, 7(13): 262–254.

Rechinger K H. 1997. Flora Iranica. Graz: Akademische Druck- u. Graz Publishing House.

Reid N, Stafford S D M, Beyer-Munzel P, et al. 1990. Floristic and structural variation in the Tamaulipan thornscrub, Northeastern Mexico. Journal of Vegetation Science, 1(4): 529–538.

Renard K, Foster G R, Weesies G, et al. 1997. Predicting rainfall erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE). Agriculture Handbook Number 703. Washington, DC: United States Department of Agriculture, 1–404.

Robles A B, Passera C B. 1995. Native forage shrub species in south-eastern Spain: forage species, forage phytomass, nutritive value and carrying capacity. Journal of Arid Environments, 30(2): 191–196.

Romkens M J M, Prasad S N, Poesen J W A. 1986. Soil erodibility and properties. In: Transaction of the 13th Congress of International Society of Soil Science. Hamburg, Germany, 492–504.

Sáez-Plaza P, Michałowski T, Navas M J, et al. 2013. An overview of the Kjeldahl method of nitrogen determination. Part I. Early history, chemistry of the procedure, and titrimetric finish. Critical Reviews in Analytical Chemistry, 43(4): 224–272.

Sandhu H S, Wratten S D, Cullen R. 2010. The role of supporting ecosystem services in conventional and organic arable farmland. Ecological Complexity, 7(3): 302–310.

Schenk H J, Jackson R B. 2002. Rooting depths, lateral root spreads and below-ground/above-ground allometries of plants in water-limited ecosystems. Journal of Ecology, 90(3): 480–494.

Schirpke U, Kohler M, Leitinger G, et al. 2017. Future impacts of changing land–use and climate on ecosystem services of mountain grassland and their resilience. Ecosystem Services, 26: 79–94.

Sharafatmandrad M, Khosravi Mashizi A. 2021. Temporal and spatial assessment of supply and demand of the water-yield ecosystem service for water scarcity management in arid to semi-arid ecosystems. Water Resources Management, 35: 63–82.

Soil Survey Staff. 1994. Keys to soil taxonomy (6th ed.). Washington, DC: United States Department of Agriculture-Soil Conservation Service.

Tallis H, Ricketts T, Guerry A, et al. 2011. InVEST 2.2.2 User's Guide. [2023-06-25]. https://naturalcapitalproject.stanford.edu/software/invest/invest-downloads-data.

Tesfa T K, Tarboton D G, Chandler D G, et al. 2009. Modeling soil depth from topographic and land cover attributes. Water Resources Research, 45: W10438, doi: 10.1029/2008WR007474.

Tongway D J, Hindley N. 2003. Indicators of ecosystem rehabilitation success. Stage Two. Verification of EFA Indicators. Final Report for the Australian Center for Mining Environmental Research. Canberra: CSIRO Sustainable Ecosystems, 1–66.

Toopchi-Khosroshahi Z, Lotfalizadeh H. 2011. Identification of honey plants and their attractiveness to honeybee in Kandovan, Northwest of Iran. Biharean Biologist, 5(1): 36–41.

Tsai C C, Chen Z S, Duh C T, et al. 2001. Prediction of soil depth using a soil landscape regression model: a case study on forest soils in southern Taiwan. Proceedings of the National Science Council, Republic of China. Part B, Life sciences, 25(1): 34–39.

Uchida K, Koyam A, Ozeki M, et al. 2020. Does the local conservation practice of cultural ecosystem services maintain plant diversity in semi-natural grasslands in Kirigamine Plateau, Japan? Biological Conservation, 250: 108737, doi: 10.1016/j.biocon.2020.108737.

Van Soest P J, Robertson J B, Lewis B A. 1991. Methods for dietary fiber, neutral detergent fiber, and nonstarch polysaccharides in relation to Animal Nutrition. Journal of Dairy Science, 74(10): 3583–3597.

Wischmeier W H, Smith D D. 1978. Predicting rainfall erosion losses - a guide to conservation planning. Agricultural handbook (United States Department of Agriculture) no. 537. Hyattsville, Maryland: USDA, Science and Education Administration.

Zhou M M, Deng J S, Lin Y, et al. 2019. Identifying the effects of land use change on sediment export: Integrating sediment source and sediment delivery in the Qiantang River Basin, China. Science of the Total Environment, 689: 38–49.